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Reducing Undesirable Environmental Impacts in the Marine Environment: A Review of Market-Based Incentive Management Measures

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Abstract

Using the example of commercial fishing, this paper explores the potential of incentive based management measures as a means of reducing the undesirable impacts of industries operating within the marine environment. Despite having been successfully applied for similar purposes in the management of terrestrial environments, and their potential to achieve environmental gains in an economically efficient manner, examples of incentive based management mechanisms are still relatively limited in the marine context.

We assess the potential of a number of alternative market based management measures by reviewing and considering the successes and limitations of previous applications and how these would translate in the case of commercial fishing. Several fishing methods and conservation values are considered and the circumstances in which incentive measures may be most applicable are identified. Where appropriate, and by either replacing or (more likely) complimenting existing management arrangements, incentive based measures have the potential to improve upon the performance of existing measures. This has a number of implications. From the environmental perspective they should allow the expected level of undesirable impact to be reduced. They can also reduce the costs imposed upon the industry by letting them develop the solutions. Further, in the increasingly relevant case of MPAs the potential costs to Government may also be significantly reduced if increasing environmental performance makes it possible for certain industry members to continue operating, reducing the necessity of often costly structural adjustment programs.

1. Introduction

The development of international (e.g. United Nations Conference on Development and Environment; the Convention on Biological Diversity; United Nations Conventions on the Law of the Sea) and regional (e.g. Convention on the Protection, Management and Development of the Marine and Coastal Environment of the Eastern African Region (Nairobi Convention); Natura 2000 (EU)) conventions to protect marine biodiversity over the last two decades has resulted in increased areas of marine habitat being closed to fishing. For example, the Convention on Biological Diversity has a global target of 10% of the marine environment being included in marine protected areas (MPA) coverage by 2012. In the US, legislation is being developed at both State and Federal level with this objective (Hildreth, 2008), while the European Marine Strategy Framework Directive[1] has a similar goal in Europe. MPAs are also being implemented in developing nations for both conservation and economic (mainly tourism and protect coastal community livelihoods) reasons (Francis *et al.*, 2009).

Australia is also currently undertaking a process of marine bioregion planning, driven by a commitment under the *Environment Protection and Biodiversity Conservation Act 1999* to implement a National Representative System of Marine Protected Areas (MPAs). As part of the planning process Australia's marine zone has been divided into five separate marine regions (the South-west, North-west, North, East and South-east), each of which is currently undergoing assessment to determine where marine protected areas will be implemented in each region. Within Australia, designation of MPAs is generally accompanied by some form of fishing effort reduction, usually through a buy-back program, and has in the past also involved compensation for related industries. Compensation involved with the expansion of the "no take" zones within the Great Barrier Reef marine park from 4% to 34% has been estimated to have been in excess of \$70m,[2] much of which was paid to onshore businesses that claimed to be adversely affected by the increase in the no take zone. The licence buyout component of the compensation is calculated to have been \$31.8m (FERM, 2007). Consequently, identifying policies that can reduce these costs, whilst still achieving management goals, is an important component of developing a cost-effective approach to marine spatial planning and management.

In some cases, marine conservation objectives may still be met without the need to fully remove fishing from an area. By reducing the level of undesirable impacts a fishery generates, and consequently increasing their compatibility with respect to identified conservation values, appropriate management measures may allow fisheries to continue operating within the bounds of zoned reserves. Reducing the level of displacement will allow the cost of creating reserves to be reduced. Depending upon the management objectives, reducing costs of

[1] Directive 2008/56/EC of the European Parliament and of the Council of 17 June 2008 establishing a framework for community action in the field of marine environmental policy

[2] The actual total compensation is unclear, with claims from various sources ranging from \$50m to in excess of \$300m, the latter largely coming from groups opposed to further rezoning of marine parks in Queensland (including Moreton Bay). A definitive figure has not been publically released.

adjustment may also allow protection to be afforded to larger areas of habitat or species ranges (i.e. making the process more cost-effective).

The aim of this paper is to review the potential of market-based management instruments as a means of increasing a fishery's (or other activity's) level of compatibility with respect to identified (generally non-commercial) conservation values. Market-based management measures aim to mitigate the impacts of activities by better aligning the incentives their participants face with the objectives of management. Where sufficient changes in behaviour can be achieved the need to displace activities such as fishing may be lessened or averted by allowing them to continue operating in a less environmentally detrimental manner. The potential of such measures is underexplored in this context and may prove to offer additional and complimentary management options when implementing MPAs. Examples of instances in which these measures have been applied previously are discussed, along with the factors that influence their appropriateness and effectiveness. We find the use of market based measures requires careful consideration at the fishery specific level but that a number appear to have potential application with regard to reducing undesirable impacts.

2. A (very) brief overview of fisheries impacts on marine environments

The impacts of fishing on the marine environment are well documented (e.g. Tasker *et al.*, 2000; Kaiser *et al.*, 2002). Fishing directly affects the marine habitat through contact with the seabed, as well as removing species from the environment that have little or no commercial value, but potentially considerable non-market value (e.g. iconic species such as turtles, dolphins and seabirds). As the cost of this damage is not borne by the fisher, levels of damage are greater than what may be socially optimal. MPAs limit or reduce this damage through preventing access to areas that are considered to have substantial non-market value (e.g. biodiversity value, non-consumptive use value such as scuba diving, or large populations of iconic species such as turtle nesting areas).

The specific conservation values a fishery impacts and the incidence with which this impact occurs will vary by both fishery and region. For example, in more southerly demersal trawl fisheries, the absolute number of whales or turtles caught is likely to be relatively low whereas the number of demersal fish or sharks will be higher (Table 1). As a result incidence of bycatch needs to be considered on a case by case basis.

Table 1. Example incidence of bycatch of non-commercial species by gear type

Incidence	Demersal trawl		Demersal longline	Pelagic longline		Demersal gillnet
	north	south		north	south	south
Infrequent		whales				
	seabirds	turtles		seabirds		
		dolphins				
	turtles	seabirds				
	sea snakes	seals/ sealions		turtles		seals/ sealions
	sharks	sharks	sharks		Seabirds	sharks
Frequent	other fish	other fish	other fish	sharks	Sharks	

A number of potential tools are outlined and discussed below with the objective of considering their ability to reduce the number of activities deemed as incompatible and in the process reduce the potential cost to government.

3. Market-based instruments and fisheries management measures

Market-based instruments work by creating a price (explicitly or implicitly) for the use of the non-market resource in the production process, thereby creating incentives to reduce its consumption. In the case of fisheries, these resources include bycatch of non-target species (including iconic species) as well as habitat damage. A hierarchy of potential market-based management systems is presented in Figure 1. Incentives can be created by influencing the rewards from fishing, or by placing soft constraints on fishing activities. Soft constrains, such as a bycatch quota, differ from hard constraints, such as an area closure, as fishers are able to adjust the level of their individual constraint through quota trading. Financial incentives include the use of charges, subsidies or bonds. Charges and subsidies directly affect the returns from different fishing activities (thereby stimulating behavioural change), while bonds provide an incentive for fishers to minimise their impacts through either technological measures or behavioural changes. How these outcomes are achieved is generally best left to the fishers, although some market access restrictions impose particular technologies on the fleet.

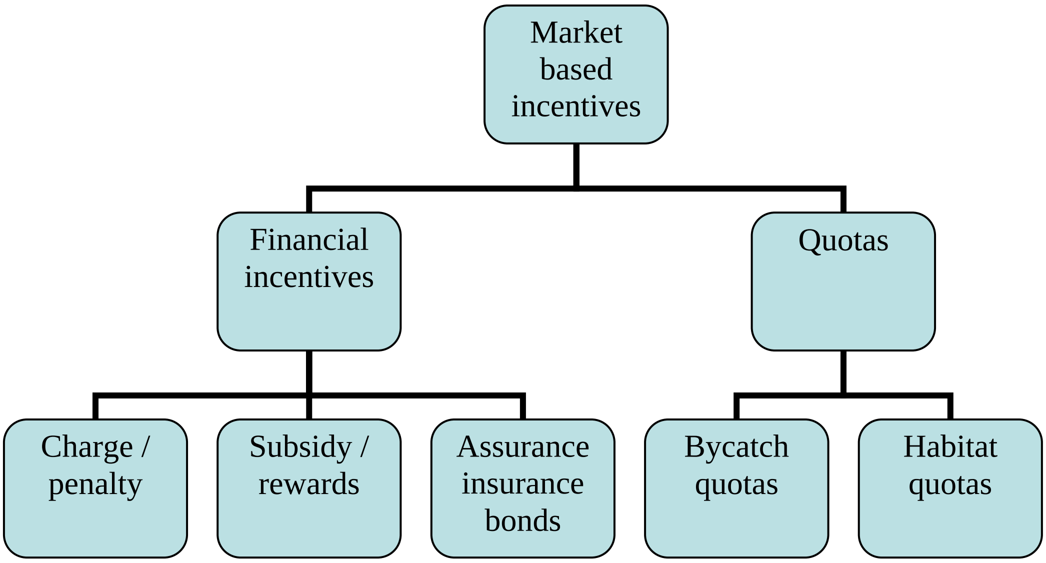

Figure 1. Hierarchy of market-based fisheries management systems for reducing environmental damage

3.1 Financial Incentives

3.1.1 Direct Subsidies

The use of subsidies to reduce bycatch and other impacts is limited in fisheries. Where subsidies exist, these are usually related to reducing the cost of fishing gear to encourage their adoption (Cox and Schmidt, 2006).

The use of subsidies in fisheries is generally discouraged, and there is international pressure to reduce subsidies in fisheries (e.g. from FAO and OECD). At a global level, subsidies to the fishing industry are one of the key factors that have led to overcapitalisation and overexploitation in many fisheries. Even "environmentally friendly" subsidies can result in increased exploitation by reducing the cost of fishing (Cox and Schmidt, 2006).

Given this, direct subsidies are not considered a desirable method for reducing an activities impact. Indirect subsidies, such as research and development of environmentally friendly fishing gears remain an acceptable option. However, these do not provide a direct incentive to fishers so are not considered an incentive based management system.

3.1.2 Charge/Penalty-based systems

Bycatch and other environmental impacts are unpriced input factors in the production process. As such, there is no incentive for fishers to limit them except through the opportunity cost they may impose in terms of time (e.g. to dispose of bycatch or clear nets fouled with seagrass) and the potential direct effect on harvest (i.e. the consumption of bait and hooks that might otherwise catch fish, or value reducing damage inflicted on the target species due to interactions with bycatch whilst in the cod end[3]). Pricing impacts appropriately provides incentives for fishers to adjust their behaviour (i.e. production and fishing effort allocation) accounting for these additional costs, and provides an incentive for fishers to adopt technologies that reduce these costs through reducing impact. When impact-reducing technologies do not exist, correctly set charges will reduce the level of production, and consequently impact, to what is deemed optimal whilst concurrently encouraging the development of impact reducing technology. For example, the use of carbon charges has been seen to influence both changes in energy mix in manufacturing and total demand by households (Johansson, 2000; Bruvoll and Larsen, 2004), and induce technological change that accelerates the substitution of carbon-free energy for fossil fuels substantially (Gerlagh and Lise, 2005).

Explicit prices for bycatch and impact can be implemented through a bycatch charge/penalty system, where fishers pay a fee for each unit of bycatch caught. The same system may be applied in the case of environmental impact where fees would be paid for each unit of e.g. habitat impacted, or numbers of marine mammals disturbed. A penalty system in this regard is a fee or levy charged by government for access to the resource (be it bycatch or habitat), and is not related to the revenue or profitability of the vessel. Fishers are then able to objectively balance the benefits of fishing in a given area or time period (i.e. the value of the retained catch) against the costs of fishing, including the cost associated with their impact due to operating there or then. An advantage of such a penalty system is that, theoretically, different impacts (and species) can attract different penalty rates thereby ensuring the greatest protection to that which is most vulnerable.

The potential benefits of a bycatch charge/penalty in reducing the level of bycatch have been demonstrated by a number of authors (Sanchirico, 2003; Diamond, 2004; Herrera, 2005; Singh and Weninger, 2009)[4] and (along with bycatch quotas) are currently being investigated within the context of the Commonwealth Trawl Sector (CTS) in south east Australia (Hutton *et al.*, 2010). These have generally considered bycatch of non-commercial species that are normally discarded, including also bycatch of megafauna (e.g. seals, turtles, seabirds). These studies cited above have largely been theoretical in nature, and examples where bycatch charges have been implemented are limited. Bycatch charges are in place in New Zealand and Namibian fisheries (detailed below), but are limited to bycatch of commercial species in a bid

[3] For example, fish becoming bruised reducing quality and therefore price, or the damage caused to target species by crabs caught in shrimp trawls.

[4] Most economic analyses refer to these charges as a bycatch tax. In effect, taxes and charges work the same way in that they place a price on the undesirable output, creating incentives to avoid it. Further, referring to such measures as taxes can result in them being considered politically unacceptable (Brown, 2000). This is because environmental taxes can be perceived as a means of transferring any economic profits generated through management to the government, so that fisher are no better off financially than under un-regulated conditions (Sanchirico, 2003).

to discourage active targeting of these species. The use of charges as an environmental management tool, however, is well established in regards to pollution (Roseveare, 2001; Bruvoll and Larsen, 2004; Andersen, 2008).

Both habitat use penalties and bycatch penalties would require information on fishing activities and catches. In the past this data been quite expensive to collect but is likely to become cheaper due to the continued development of electronic monitoring systems, these are considered further in section 4.

One possible role for exploring the more general use of bycatch monitoring and penalties in marine reserves would be as a test case for their more general introduction. Marine reserves could thus be used as experimental opportunities for improved management.

Despite the demonstrated theoretical benefits in terms of reduced bycatch, there are relatively few examples of where these measures have been applied in the management of fisheries. Further, they have been limited only to the management of bycatch of commercial species, primarily to either dissuade fishers from targeting species for which they held no quota (such as in Namibia) (Schrank *et al.*, 2003; Rukoro, 2009), or provide an incentive to land (rather than discard) overquota catch (such as in New Zealand and Iceland) (Sanchirico *et al.*, 2006). In the latter case, too low a charge may result in greater levels of catch than under a discard policy (Churchouse, 2007). However, while prohibition on the sale of bycatch (i.e. forced discarding) may reduce the bycatch level, it may also reduce social welfare (Boyce, 1996).

A penalty based system may be more effective for non-commercial species as no direct incentive to target them exists, other than the value of the associated targeted species. Where fishers are able to avoid the non-commercial species, a bycatch charge is likely to influence their behaviour and reduce the catch of these species. As with many of the policies discussed in this review, however, the effectiveness of the market based instrument will depend on the actual ability of the fisher to avoid the species of concern.

3.1.3 Assurance Bonds

Assurance, or performance, bonds are economic instruments commonly used in environmental management (Shogren *et al.*, 1993; Cornwell and Costanza, 1994; Ferreira and Suslick, 2001; Bagstad *et al.*, 2007). In most applications, assurance bonds require the user of the resource to place a sum of money deemed equivalent to the potential damage that the activity can have on the environment as a security. This bond is refundable provided the damage is not incurred, or is repaired by the resource user (e.g. through offsets or habitat restoration work).

Although such a system has not yet been applied in commercial fisheries management, it is possible to envisage how it may be employed. In the context of fishing, the instrument would require the fishers to place funds into a trust. These funds would be returned (with interest) provided the fishers achieved a pre-determined performance target in terms of bycatch rates or avoidance of habitat impacts. Such a system could operate at either the individual level or at the level of the fishery, with each fisher having joint and several liability for the actions of the group as a whole. This latter option creates incentives for self-regulation and fosters collaboration in terms of information sharing (e.g. how to avoid the bycatch). However, it may also create perverse incentives: if the individuals believe the fishery target will be exceeded, they have no individual incentive to reduce their own bycatch and incur the additional costs in doing so. As they would have already contributed to the fund and believe these to be lost, they have no incentive to incur additional costs by bycatch avoidance. This is effectively the prisoner's dilemma – it is in everyone's interest to collaborate, but if trust regarding the behaviour of others is absent, then it is in each individual's interest not to collaborate (Tucker, 1950). Once the trigger level has been exceeded and the bond is lost, there is no incentive for any further bycatch reduction. Near real time monitoring and management would be necessary to prevent this eventuality. Whilst such perverse incentives are a potential problem with any industry, it is arguably more likely within fisheries as the number of firms participating is generally higher than in others such as mining.

Even if individuals initially believe that others are reducing their environmental damage, there may still be incentives for some individuals not to conform. As avoiding bycatch results in increased costs to the fisher, some will choose to free-ride on the behaviour of others and not reduce bycatch. Eventually, this behaviour will create the belief that the bond is lost, with bycatch increasing again. Consequently, although a fishery level assurance bond has the potential to be self regulating, the incentives to the individual not to comply may result in this being ineffective as a bycatch management option in some instances.[5]

Fishery level assurance bonds may only be viable in relatively small fisheries, where individuals are aware of each other's activities and are able to influence the set of behaviours. In such fisheries, self regulation may be possible as the cost incurred by each individual is effectively affected by the behaviour of others. This is likely to encourage information sharing as well as innovation sharing within the group. In larger fisheries, free riding, and expectations of free riding, are likely to result in the bond effectively being a fixed cost of operation, or a fishery closure depending on the size of the bond.

By contrast, the use of assurance bonds levied at the individual fisher level may be less complex. For instance, in the context of marine zoning access to different areas of the fishery could be subject to different bond levels depending on environmental sensitivity. An advantage of individual bonds in this situation is that individual fishers could choose to either pay the bond to access a particular area or fish elsewhere. The bond provides an incentive to either adopt technologies to minimise the chance of violation (if operating in the bonded

[5] Social based incentives, however, could develop as fishers complying with the program would put pressure on potential non-compliers in order to protect their share of the bond.

area), or to avoid the sensitive area entirely. Less fishing in the bonded area also reduces the likelihood of the adverse environmental impact, and may also provide additional benefits to fishers in terms of higher catch rates due to lower competition (at least in the short run).

A difficulty with both individual and group bonds, as with bycatch charges, is that there are no incentives for fishers to accurately report their level of damage, and widespread surveillance coverage would be necessary. Electronic surveillance systems may be suitable for such a task and, again, are discussed further in section 4. With individual bond systems, vessel monitoring systems (VMS) also enables identification of which vessels are fishing in a bonded area, and when they are fishing in these areas.

A further difficulty with bond-based systems is that, given the potential magnitude of the financial penalty at the fishery level, legal challenges are likely to emerge that could slow down (if not hamstring) the process. Ideally, the size of the bond, and the magnitude of any forfeiture, will be based on the potential value of the damage (or the cost of reparation). Determining these will be difficult, and behavioural changes as a result of the bond itself will potentially alter the level of impact. Given this uncertainty, actually recovering the bond may be difficult and costly.

For this reason, bonds may be best related to habitat damage rather than bycatch as the latter is more uncertain than the former. Estimates of habitat damage may be derived from monitoring the amount of time fished in an area a particular type of gear. Fishers can choose which areas to fish in within a reserve, and which gears to use (when more environmentally friendly gear options exist) to minimise this damage. Restoration may involve excluding vessels from the whole or part of a reserve if damage limits are exceeded, and the cost of this (in terms of compensation to fishers conforming to the requirements of fishing in the reserve who are subsequently displaced) can be readily estimated. In contrast, bycatch of iconic species is generally an infrequent event, with a highly random component. While vessels can use gear that reduces the likelihood of this event, their ability to fully avoid bycatch is beyond their control. Similarly, rehabilitation costs (in terms of stock rebuilding) are also difficult to value, so claiming damages will be more complex.

Some examples of the use of bonds in aquaculture and marine reserves exist. Commercial operations that take place within the Great Barrier Reef Marine Park generally require a permit. For non-fishing businesses (particularly dive and recreational fishing charters), this generally necessitates entering into a Deed of Agreement that binds the permit holder to certain obligations that can include; indemnifying the GBRMPA, maintaining adequate insurance, ensuring removal of structures and clean-up of the site of operations if directed and the payment of a bond (or bank guarantee of similar value) for structures other than vessel moorings (GBRMPA, 2009) (Smith *et al.*, 2005). These funds have been accessed on a number of occasions to remove abandoned equipment from tourism and pearl aquaculture (ABARE, 1993; Lal and Brown, 1996; Smith *et al.*, 2005).

The Aquaculture Lease Security Arrangements imposed on oyster farmers in the Australian state of New South Wales are also bonds, designed to ensure the industry shares the responsibility in the future for problems arising from any lease management and maintenance issues.[6] Similarly, bonds have been used and are increasingly common as a tool for ensuring proper decommissioning of offshore oil and gas facilities (Ferreira and Suslick, 2001). Experience with bonding suggests that these applications could be extended to other economic activities where modifying ongoing behaviour is the target, as opposed to remediation of past impacts.

3.1.4 Insurance

An alternative to an assurance fund would be *insurance*, where those involved in an activity would contract to undertake financial liability. The key difference between an assurance and insurance bond is that with the former, the funds need to be provided up-front before the activity (e.g. fishing) can take place, whereas in the latter the industry only need to raise the funds in the event that the performance is not achieved. Both effectively represent a fine for non-performance. A potential benefit of an insurance-based system is that the risk could be potentially sold on the insurance market, with industry members paying a premium to the insurer.

Where the insurance market is used to spread the risk, the premiums paid would reflect the insurer’s perception of the probability of liability, which in turn will be influence by the insured’s past performance and adoption of mitigation technologies. Poor performers in terms of the specified impact would most likely incur a higher premium, while adoption of mitigation technology may attract a discount. This would provide additional incentives for individuals to modify their behaviour.

Such schemes are likely to be most effective when the chance of an impact is relatively small, and is highly observable. For example, bycatch of turtles and marine mammals are more readily observable, and potentially more avoidable than general bycatch of non-commercial species.

The aim of insurance is both to compensate and deter. Insurance markets have been used in the management of pollution in a number of countries, including Australia (e.g. the NSW Contaminated land Management Act of 1997) (OECD, 2003). Experiences with insurance markets for environmental management in other sectors suggest that insurance markets are more effective when restoration costs are recovered rather than a fixed penalty representing the value of the environmental damage. This is largely a legislative issue, as defining an

[6] www.dpi.nsw.gov.au

economic value of environmental damage for compensation to society is more complex[7] than an unspecified restoration cost. In the case of marine animals, restoration implies either some form of compensatory mitigation (i.e. offset programs) or, in some cases, fishery/area closures to allow recovery. The cost of these will vary on a case by case basis, and the insurance market will determine the appropriate premiums to reflect both the risk of occurrence and the likely magnitude of the costs.

A potential problem with insurance, however, is the moral hazard problem. That is, once insured, fishers have no immediate incentives to avoid the bycatch. This problem is addressed through more common insurance systems (e.g. car insurance) though imposing partial contributions from the insured party in the event of a payout and through initial high premiums, with discounts for mitigating measures being taken and also “no-claims” discounts. The incentive to avoid is then maintained even though the fisher is insured, as a cost (in terms of higher current and future costs) is imposed if bycatch is taken.

3.2 Quota Systems

These are typically employed on the basis that when the quota is reached the vessel or fishery in question must cease operating for the remainder of that season/period. A number of quota based systems have been proposed and this section considers those based on bycatch, revenue, effort and habitat.

3.2.1 Bycatch Quotas

Bycatch quotas may be applied either at the fleet level (i.e. aggregate bycatch quotas) or the individual vessel level. As the incentives created differ depending on which level is implemented, the approaches are considered separately below.

An advantage of quotas is that in only setting the permissible level of bycatch (whilst conforming to any pre-existing regulation), it leaves open the method/s by which preventing a closure may be achieved. In doing this, it overcomes the criticism often levelled at more command and control orientated management methods, such as the mandatory application of technical measures (e.g. TEDs), that ‘one size’ does not ‘fit all’ when local circumstances or regional variations possibly render them ineffective or inappropriate. Typically such measures may be expected to encourage fishers to move out of areas in which high rates of bycatch are likely, allowing them to continue operating, or propagate improvements in gear so that less bycatch is taken. A further advantage is that in specifying an exact number, a clear and easy to understand goal is set and can be worked towards, even if the level at which it is set is disputed (Bache, 2003).

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[7] This is a different issue to setting a fixed penalty for bycatch, as the aim of the penalty is purely to dissuade an activity, whereas the principle of the insurance is that it covers the cost of damage as compensation.

Aggregate bycatch quotas

To date, examples of bycatch quotas are limited, and these have generally been aggregate quotas rather than individual quotas. Aggregate quotas contain characteristics more akin to command and control management measures as they are a hard rather than soft constraint to fishing. However, where they have been applied they have often generated incentives to reduce bycatch (Bache, 2003; Gilman *et al.*, 2006; Haynie *et al.*, 2009).

Aggregate quotas limit or reduce bycatch by capping the total permissible level of bycatch over a specified period of time. Once the threshold level is reached a fishery may be closed for the remainder of the season. Such bycatch quotas can be adjusted over time to reflect the state of the bycatch species stock or gradually reduced if the aim is to encourage vessels to become more efficient in this respect. This method of regulation will only propagate improvements in performance if bycatch quotas are set at levels that are likely to be limiting under current conditions (and consequentially result in a loss of revenue once reached). They will also only improve performance down to the level at which the cap has been set.

There are a number of cases where bycatch limits for non-target species are imposed on fisheries, although these have mostly been related to bycatch of megafauna. In the US, a potential biological removal rate (PBR) is defined for several fisheries, being the maximum numbers of marine mammals that may be taken by fisheries. This is calculated annually and is designed to ensure few enough animals are taken so that stocks do not fall below levels considered to be optimally sustainable.[8] Similarly, a total allowable catch of turtles (along with a number of technical measures) was introduced into the Hawaii based long-line swordfish fishery was closed completely over the period 2001 to 2004 due to concerns over excessive levels of turtle bycatch (US National Marine Fisheries Service, 2004). However, a spike in the level of demand for swordfish in 2006 resulted in a “race-to-fish”, with a large increase in the number of hooks set early in the year and the fishery being subsequently closed early (Gilman *et al.*, 2007).

An annual total level of dolphin catch is set under the Agreement on the International Dolphin Conservation Program (AIDCP) for vessels operating in the eastern tropical Pacific (ETP) purse seine fishery. This is divided between the states and then vessels taking part in the fishery, but as a result of limited entry and other regulations pertaining to this fishery effectively limiting bycatch the TAC is not reached.[9]

New Zealand also uses output controls to manage bycatch of Hooker’s sea lions in the arrow squid trawl fishery (Bache, 2003; Diamond, 2004; Chilvers, 2008). Between 1996 and 2007,

[8] The optimal sustainable population level (OSP) being defined as the number of animals which will result in the maximum productivity of the population or species.

[9] A total limit of 5000 dolphins was set for 2006 with actual mortality estimated at 886 for the year (IATTC, 2008). During 2007, only 6 per cent of all sets made on tuna associated with dolphins involved mortality or serious injury to the dolphins. The total mortality of dolphins in the fishery has been reduced from about 132,000 in 1986 to less than 900 in 2007 (www.iattc.org/DolphinSafeENG.htm).

the fishery has been closed early six times out of 12 fishing years, with two other years initially closed but overturned by the Court of Appeal (Chilvers, 2008). The bycatch quota management system has resulted in the industry investing heavily in ways to reduce their rates of sea lion bycatch (Bache, 2003). This resulted in the development and application of a sea lion excluder device (SLED), similar in concept to the TED, as well as increased sharing of information amongst vessels to reduce sea lion bycatch (Diamond, 2004).

In Australia, aggregate quotas on seabird bycatch are currently in place as part of a seabird bycatch threat abatement plan relating to bycatch during oceanic longline fishing operations. Trigger rates (on the basis of the number of birds caught per thousand hooks deployed) are set for several different. Once these trigger levels are exceeded in an area, the area is closed. This trigger has been exceeded in the pelagic longline fishery, which has been closed to daytime fishing twice in the last couple of years.

Individual transferable bycatch quotas (ITBQs)

As noted previously, a main problem with bycatch is that it has no value to the fisher, and is subsequently discarded. Further, its capture and subsequent discarding results in minimal cost to the fisher also. A cost can be created through a charge or penalty system (as discussed above). However, a cost can also be created through allocation of quotas for the bycatch species. Several authors have suggested the use of individual transferable bycatch quotas (ITBQs) as a means of reducing bycatch (Boyce, 1996; Edwards, 2003; Diamond, 2004; Herrera, 2005; Hannesson, 2009; Ning *et al.*, 2009). ITBQs improve efficiency in a fishery by creating a “shadow price” associated with use of the quota reflecting the level of economic rents derived from the use of the quota. As such, the shadow price reflects the value of the targeted species caught with the bycatch species rather than the bycatch species *per se*. This shadow price, while different for each individual, manifests itself in the form of a quota trading price. Fishers’ with a shadow price greater than the trading price are likely to buy quota, while those with lower shadow prices are likely to sell. Given the shadow price reflects the catch compositions of the vessels, this guides quota to those boats that can use it most efficiently. By creating a quota market for bycatch, fishers have an incentive to lower their own bycatch and sell their quota to fishers less able to lower bycatch, fostering innovation and adoption of new technologies.

ITBQs have been suggested for both megafauna (Hannesson, 2009; Ning *et al.*, 2009) as well as fish species – either commercial (by-products) or non-commercial (Boyce, 1996; Diamond, 2004). However, the evaluation of these programs have been model based rather than experience based as, to date, such programs have generally not been implemented. These analyses generally ignore enforcement issues (Boyce, 1996), which are likely to be considerable.

Relatively few real life examples of ITBQs can be found, and those that have are focused on bycatch of commercial species. In 1996, Canada instituted an individual vessel bycatch quota

(IVBQ) for its groundfish trawl fleet, which helped reduce total fleet bycatch from 681 mt in 1995 to 140 mt in 1996 (Diamond, 2004). These quotas were non-transferable, and once the vessel reached its bycatch quota for a particular area, fishers would need to either cease fishing or move to an area where it had available bycatch quota. Similarly, several shark species caught as bycatch are included in the NZ quota management system. These species have a commercial value, but are generally caught as bycatch in other fisheries. A recent international review of the status of shark stocks suggests that, while these species are globally considered as vulnerable, the stocks managed in NZ under the individual quota system are either healthy or recovering (Camhi *et al.*, 2009).

3.2.3 Individual Habitat/Spatial Effort Quotas

An alternative form of effort control is the individual habitat quota (Sanchirico *et al.*, 2006). These are spatial management instruments where different effort penalties are applied to different areas based on the level of damage created by fishing in those areas. These quotas are tradable, allowing vessels to plan and adjust their fishing activities to minimise their own damage. Fishers consume their quota based on where and when and how they fish, with the penalty system providing incentives to either operate in areas where less damage will be incurred, or adopt fishing gear that will have a lower impact. Ideally, such a system would impose differential penalties based on gear used. Such a system provides an incentive to either reduce effort, or the use of more environmentally friendly gear, in sensitive habitats without the need to impose a total closure.

While not designed with bycatch in mind, such a system can also be adapted as a bycatch management system. Indeed, the hook decrementation system introduced in late 2009 in the Australian Eastern tuna and billfish fishery (ETBF) is effectively an example of such a system (Pascoe *et al.*, 2009). Individual fishers have an individual hook quota. The rate at which this quota will be consumed depends on where and when they fish. Areas with high bycatch of species of concern (mainly seabirds, turtles and sharks) could attract a high penalty rate, whereas other areas with little bycatch could attract a much lower rate. This system also allows management to be responsive by looking at the outcomes of landings and adjusting spatial decrements accordingly, thereby manipulating fishers' realised spatial effort quota.

A requirement of any such quota management system is that the 'optimal' level of impact to be permitted (i.e. damage or bycatch) or stock to be preserved (in terms of either habitat or biota) needs to be pre-determined in order for the quotas to be set.

To date, no examples of use of such a system are available to evaluate its effectiveness. The ETBF hook penalty system has only been operational for 3 months, so is too early to determine how fishers have responded to the incentive system.

4. Discussion

Whilst charges, bonds and quotas can all create the necessary incentives to reduce impacts, alone they are not sufficient as in the absence of monitoring and enforcement there is no incentive to report any impact caused. In fact, if reporting bycatch or habitat damage is likely to result in some form of penalty there is an incentive to conceal these events.

Historically, the reliable detection of bycatch or habitat damage has been relatively expensive due to the need for independent observers on each vessel during fishing operations. However, recent technological changes[10] are potentially revolutionizing the use of these types of measures. In Europe, video monitoring systems have been successfully trialled aboard Danish whitefish vessels where they were found to be both highly accurate and substantially cheaper than observer coverage (Dalskov and Kindt-Larsen, 2009) and a number of Scottish whitefish trawlers are currently taking part in trials. Further, vessel monitoring systems which track vessel locations are now common among the larger fisheries at both state and Commonwealth levels, and can be used to monitor habitat impacts by examining overlap with vulnerable habitats. Similarly, Australian Fisheries Management Authority AFMA is implementing video monitoring systems in some of the major fisheries, including the pelagic longline fishery and the northern prawn fishery. These systems record all fishing operations on the vessel, and data can later be collected on catches and operating conditions. In the pelagic longline case the video will be used to validate the existing logbook data program. Operators must record fishing practices and catches in a logbook. Ten percent of video for each fishing vessel will be examined and checked against the logbook records. Any discrepancy will result in the operator paying for the full annual record to be matched to the logbook, and the operator will be subsequently fined for any violations detected. This policy is likely to provide a very strong incentive for accurate data recording, and can be expected to provide records of a sufficient quality for use in incentive programs.

An additional factor for consideration when using market-based measures to reduce the level of undesirable impacts caused by fishing is the incidence with which any damage is expected to occur. The incidence of occurrence influences which measures are likely to be most practical and effective and is a factor that varies by both fishery and species (Figure 2).

[10] E.g. video recording each haul; weighing each haul and determining discards as the difference between total landed and caught weight, etc.

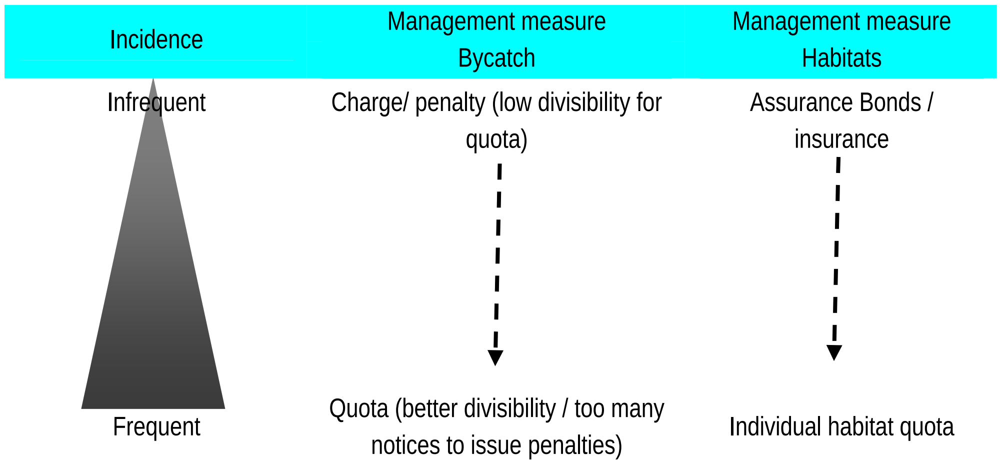

Figure 2. Relationship between frequency of occurrence and appropriate market based instrument

For bycatch species that are likely to be taken on a relatively infrequent basis a charge/penalty based system is likely to be the most appropriate method of incentivising further reductions in levels of occurrence. This is also a good measure for managing the bycatch of multiple species as a suite of charges can be applied, with the level of each charge set to reflect the importance of that species. It is flexible and can easily be adjusted to meet the needs/aims of management. If desired, penalties can also be implemented that increase with increasing levels of bycatch, either progressively or once certain thresholds are reached. In doing this, minimum levels of bycatch may effectively be realised at least cost to the industry. Fishers who aim to reduce bycatch but occasionally catch some (due to the stochastic nature of the activity) will receive generally low penalties, while those who do not take evasive action will end up with higher penalties). While theoretically applicable to all the bycatch listed in Table 1, penalties are more practically applied to species that are likely to be caught in relatively low volumes, such as turtles. The transactions costs involved in repeatedly collecting small payments for species frequently caught as bycatch may be considerable. An annual reconciliation may be more practical, but may not create the same incentive as the costs are not imposed at the time of the catch.

For bycatch species that are likely to occur on a more frequent basis individual bycatch quotas may be more appropriate. As a predetermined bycatch allowance would be allocated or purchased in advance the problem of issuing multiple penalties *ex post* is circumvented. Furthermore, levels of quota for more frequently taken bycatch will likely be larger and consequently have higher divisibility, making trade between vessels more probable and enable vessels to buy extra quota if necessary. Conversely, for less frequently caught bycatch species (or those for which frequent catches are unsustainable) only small quantities of bycatch quota would be issued and may result in problems if vessels unexpectedly catch a species and then cannot obtain quota. Conversely, fishers who hold quota may be unwilling to release it if there is a chance that they may catch some of the regulated species. If the fishery in question takes a mixture of high and low frequency bycatch species a combination of both bycatch quotas and charges/penalties could potentially be simultaneously applied.

Both charges/penalties and bycatch quotas require some form of surveillance if they are to be effective. Observer schemes can be costly, and although it is possible to require industry meets this cost, the use of video surveillance – currently being trialled in a number of fisheries – may be a practical requirement for such systems to be feasible. The benefits of this are potentially twofold in that it has the potential of both reducing costs and allowing vessels too small to physically accommodate an observer to effectively demonstrate compliance (and thereby operate in certain areas).

A further consideration if setting penalties to reduce the level of undesirable impacts is the level at which these must be set. If set too low the management objective will not be achieved but if set too high additional costs may be imposed upon society. Economic theory relating to optimal levels of pollution (e.g. Baumol, 1972), in which the marginal cost of abatement is considered in relation to the marginal damage imposed, can be applied. By accounting for the relationship between these factors the level of penalty required to incentivise a given amount of impact reduction can be determined. An example considering the concept of optimal levels of discarding in fisheries, and the costs imposed, is discussed in Pascoe (1997).

For marine habitats, the use of individual habitat quotas (HQ) may be the most effective means of limiting damage in marine reserves (other than complete exclusion). An attractive feature of this measure is that compliance can be easily assessed using VMS data, especially in combination with a video system that monitors fishing activity. A key challenge is to determine the total level or area of impact deemed as acceptable over any given period of time (e.g. season/year/indefinitely). If the ultimate aim is to progressively reduce aggregate impact the total level of permissible impact may then be reduced over time so that fishers must either apply less effort in that area or become more environmentally efficient (e.g. via the development of gears that result in lower levels of impact per unit of effort applied). Variants of this type of spatially related effort measure may also be applied to tackle bycatch when the areas in which the bycatch occur are discrete and do not overlap the majority of the target species distribution. A limitation to the gradual implementation of habitat quotas is in low energy environments, especially the deep sea, where habitat regeneration times may be measured in decades or centuries rather than years.

An alternative to habitat quotas is the use of bonds/insurance that are forfeited, or must be claimed, if acceptable (predetermined) levels of impact are exceeded within a defined period (again typically a season or year). In this instance forfeiture would need to be based upon areas being assessed on an ongoing basis, possibly making their use less attractive than habitat quotas from the management perspective.[11] In such a situation, the level of a bond could equate to the cost of replacing seagrass beds, the cost to the rest of the fishery due to these grounds being closed for *x* period of time, or both. Bonds may be more feasible than individual habitat quotas where the critical impacts are likely to occur in a relatively small area (geographically), and controls outside these areas are not deemed necessary for conservation purposes.

5. Conclusion

For the fisheries examples examined above, the options that offer the greatest potential as general management measures are a system of charges/ penalties and individual quotas (habitat and bycatch quotas). For these to be effective, monitoring systems will need to be developed that will enable catch of each species to be quantified remotely. Video monitoring systems capable of achieving this are already being trialled in several fisheries internationally and within Australia. Similarly, VMS are currently in place in most Australian fisheries, and continuing development of these technologies will enable habitat quota systems to be implemented effectively. Where access to a particular area is of concern (e.g. activity within an MPA), then bonds and/or use of insurance markets may also be of particular benefit.

[11] Effectively, the same information required for a habitat quota system (i.e. VMS with information on damage rates per day/hour fished per area) would be required to assess the status of the area. However, the responsibility for monitoring and assessing damage under a bond system would lie fully with the manager, whereas fishers have an incentive to monitor their own damage levels – and take avoidance action where possible – under an individual habitat quota.

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